CHAPTER 4: RECOVERY GOALS

Introduction

The goal of this recovery plan is to establish a framework for the recovery of the Sacramento River winter-run chinook salmon population through a logical program of improving the habitat and environment of this species. Specifically, the recovery of this species requires actions which increase their abundance and improve their habitat to the point that the probability of subsequent extinction will be very low. When the underlying causes of the species' decline are no longer in effect and the species has rebounded to relatively healthy levels, winter-run chinook can be removed from the list of threatened and endangered species; that is, it can be "delisted".

At every stage of implementation of a recovery plan, monitoring is necessary to verify whether the strategies are working. Typically, the "bottom-line" indicator of success in restoring an ecosystem upon which a species depends is certain numbers of individuals observed over an extended period of time. By giving a target number and specifying how many years those numbers have to show up in the system, environmental fluctuations that may affect populations can be accommodated. These numbers are expressed as "delisting criteria". According to the Endangered Species Act, these are "objective, measurable criteria which, when met, would result in a determination...that the species be removed from the list".

In this section, delisting criteria are presented for the Sacramento River winter-run chinook salmon population. The criteria includes two components, both of which must be met for delisting: (1) population growth rate (also termed cohort replacement rate), and (2) a numerical escapement goal. These numbers are indicators that the strategies for recovering the population have worked and the population has reached a level of sustained natural production. Although artificially produced fish may be used to supplement rebuilding of the population, direct satisfaction of the criteria will depend upon natural reproduction.

The delisting criteria were developed with the assistance of a winter-run chinook salmon extinction model. The choice of delisting criteria involved a realistic accounting of the uncertainties associated with meeting such criteria. When determining whether the population has satisfied the delisting criteria, the values of population parameters such as the mean cohort replacement rate will not be precisely know, but will be estimated from population data with the attendant error. These errors were accounted for in developing the delisting criteria to ensure a sufficiently low probability of extinction. Explicitly accounting for uncertainty is not common practice in viability analysis, but it is necessary for an accurate assessment of risk.

Delisting Criteria

The delisting criteria for the Sacramento River winter-run chinook salmon population are summarized below. An explanation of the criteria follows in the section entitled Extinction Model. The analysis which led to the delisting criteria is discussed in detail in Appendix 3.

Population Criteria

1) ***The mean annual spawning abundance over any 13 consecutive years shall be 10,000 females.[1] The geometric mean of the Cohort Replacement Rate over those same 13 years shall be greater than 1.0. Estimates of these criteria shall be based on natural production alone and shall not include hatchery-produced fish. The variability in Cohort Replacement Rate is assumed to be the same as or less than the current variability.***

2) ***There must be a system in place for estimating spawning run abundance with a standard error less than 25% of the estimate, on which to base the calculation of the population criteria. If this level of precision cannot be achieved, then the sampling period over which the geometric mean of the Cohort Replacement Rate is estimated must be increased by one additional year for each 10% of additional error above 25%.***

Recovery goals must ensure that natural populations are large enough to avert the risks associated with small population size. The numeric goals described above provide the means of assessing whether the winter-run chinook population has reached a viable, self-sustaining level. Accordingly, both the natural cohort replacement rate and spawner abundance must be evaluated. This is because a high replacement rate with few parent spawners does not necessarily indicate recovery of the population. Conversely, an abundant spawning population may not indicate a recovered population if the cohort replacement rate was negative (i.e. a declining population).

The Cohort Replacment Rate (CRR) is a parameter used to describe the number of future spawners produced by each spawner. This spawner-to-spawner ratio is defined as the number of naturally produced and naturally spawning adults in one generation divided by the number of naturally spawning adults (regardless of parentage) in the previous generation. As such, the ratio describes the rate at which each subsequent generation, or cohort, replaces the previous one, and can be described as a natural cohort replacement rate. When this rate is 1.0, the subsequent cohort exactly replaces the parental cohort and the population is in equilibrium, neither increasing or decreasing. When the rate is less than 1.0, subsequent cohorts fail to fully replace their parents

[1] Because the specified spawning abundance is in terms of females, the total spawning run will be more than twice the female spawning abundance.

and abundance declines. If the ratio is greater than 1.0, there is a net increase in the number of fish surviving to reproduce naturally in each generation and abundance increases.

For the winter-run chinook, this parameter has varied from year to year, but for the most part, values have been less than 1.0, as expected in a decreasing population (Figure IV-1). In the future, environmental and habitat conditions will have to be improved enough for these values to be greater than 1.0 to rebuild the population. CRR must then remain at least near 1.0 for a period of time at high abundance to consider the species delisted. When estimating the value of CRR, the true value of CRR will not be known. Hence, a certain number of samples will be needed to obtain an adequate precision. To adequately estimate CRR for winter-run chinook, the number of samples necessary is 9, which requires 13 years of observation of spawner abundance because the maximum spawning age is 4 years.

However, these population criteria assume that spawning abundance can be estimated with a precision of 25% (i.e., a standard error of 25% of the value). Currently, the precision in spawning run estimates is low. Estimates are based on sampling at RBDD over only part of the season and have an approximate percentage error of a little over 100% (with a one standard error range of 44% to 230%). Because a standard error of 25% may not be achievable, we have included specifications for that eventuality in our delisting criteria above. Specifically, that the sampling period must be increased by one additional year for each 10% of additional error above 25%. Also, the appropriate measure of variability in CRR impled here is the variance of the natural logarithm of CRRs.

Extinction Model

The model used to determine the probability of extinction of winter-run chinook salmon under various conditions was a discrete-time, age-structured model (e.g., Caswell 1989). The fraction of the population spawning each year was based on returns of tagged winter-run chinook salmon over 3 brood years (Hallock and Fisher 1985). In that study, the average returns were 25% as 2-year-olds, 67% as 3-year-olds, and 8% as 4-year-olds. Assuming an overall sex ratio of 1:1 and that no returning 2-year olds are females, the fractions of males returning at ages 2, 3, and 4, are 0.50, 0.44, and 0.06, while the fractions of females are 0.0, 0.89, and 0.11, respectively. Assuming there are enough spawning males to fertilize the eggs of all of the females, the model only needed to keep track of females. Fecundity was assumed to be independent of age because the dependence of fecundity on age and size is not known. The fraction of a cohort's spawning occurring at each age was therefore taken to be the fraction of females derived from Hallock and Fisher (1985).

The model describes recruitment in each year as the sum of the relative amount of spawning by spawners in past cohorts, multiplied by the fraction that survives from egg to recruitment, the Cohort Replacement Rate (CRR). Recruitment is defined to occur just after entry into saltwater and environmental variability is assumed to occur before that age. In the construction of this model, it was assumed that there was no density-dependence. Density-dependence in salmon populations typically occurs in reproduction, but in this case the limiting resource (spawning habitat) appears to be adequate. There was no evidence of density-dependence in the spawning abundance data, rather the population declined geometrically indicating a low constant survival.

The dynamic behavior of the population and the probability of extinction will depend on the distribution of these CRRs. This distribution was obtained from spawning escapement data from the Red Bluff Diversion Dam. The model was fit to these data using the age distribution of spawning females from Hallock and Fisher (1985). The resulting values of CRRs are shown in Figure IV-1, and the resulting fit to the escapement data is shown in Figure IV-2. The fact that most of the CRRs are less than 1.0, the value required for a self-sustaining population, indicates they come from a declining population. Note that these values produce a time series of spawners that is very close to the actual time series.

The probability of extinction within 50 years for this population, is based on use of the historical CRRs (Figure IV-2) in Monte Carlo simulations of the model. Extinction was defined to have occurred when the abundance of all three of the three main spawning population (i.e., those characterized in terms of fish spawning at age 3 falls below 100 females within 50 years.

Describing extinction as falling below a threshold (rather than as going to zero) is referred to as quasi-extinction (Ginzburg et al. 1982). This approach reflects the unknown deleterious effects that take place at low abundance, such as depensatory Allee effects (Allee 1931, Dennis 1989) and others. For example, when the number of spawning females is below 100, it is possible that individuals would have difficulty finding mates and the effects of demographic stochasticity would increase.

In an attempt to detect depensatory effects at low population levels of a number of fish species, Myers et al. 1995 found depensation in only a few. Among these populations were several salmon stocks, and in the most convincing case, depensation occurred at 100 females. The spawning distribution of winter-run chinook is relatively broad, where temperature conditions for successful spawning can extend for 57 miles of this wide river. With this broad distribution, a quasi-extinction level of 100 females is considered reasonable. Also, the genetic effects of inbreeding tend to be exacerbated when the number of females in the spawning run drops below approximately 100 (see the section on Genetic Considerations below).

Development of Delisting Criteria

This extinction model was used to develop the delisting criteria that would ensure a low probability of subsequent extinction once the criteria have been reached. The risk level chosen was a probability of less than 0.1 within the 50 years following delisting. This risk is less conservative than some levels used for other species, but it seems adequate given: (1) our consideration of parameter uncertainty is more conservative than other approaches, and (2) this species will continue to be monitored after delisting.

Assurance of the probability of extinction required specification of the population growth rate in addition to population abundance. Many recovery plans specify only abundance as a recovery criterion. However, there are problems with this approach because endangered populations can be increased to specified abundances without the improvements in habitat being accomplished that will allow them to remain at such high abundances. This is especially true of salmon which can be easily produced artificially. Because of this we included specification of the cohort replacement rate (CRR) for the population as a direct indication of the quality of the habitat. We chose a geometric mean of 1.0 as the desired average value of CRR, and assumed that variability about that mean would be equal to or less than the current value.

To determine probabilities of extinction at various spawning abundances, the population was simulated with the geometric mean of the CRRs increased to near 1.0 to reflect improvements in habitat. This allowed us to choose the value for spawning abundance that produced a probability of extinction less than 0.1 in 50 years. In doing this we had to account for the fact that when attempting to delist this population, managers will not know the true value of the average CRR, but will have to estimate it from estimates of spawning abundance over several years, either by counts at RBDD or by other means. The more uncertain managers are of the observations from the population, the longer they will have to sample the population to be confident that the probability of extinction in 50 years is less than 0.1. The way in which we accounted for these uncertainties is best explained by starting with the results that do not account for uncertainty, then seeing how these results change as we add a specific account of sampling error and estimation error. If the average CRR was known with complete certainty, the probability of a population of 10,000 females going extinct would be 0.0024, far below our desired probability of 0.10.

Error in Estimating Average CRR

We cannot use this result in the delisting criteria directly, because it does not account for the uncertainty involved in estimating the average cohort replacement rate (CRR). In the future when managers are attempting to determine whether the winter-run chinook salmon population can be delisted, the geometric mean of the CRR will not be known exactly. It will need to be estimated from several observations of annual spawning abundance, and these will vary in response to varying environmental conditions. Because there is error involved in the estimate of average CRR, there is a higher probability of extinction than if we knew the average of the CRRs exactly. The precision of this estimate will increase as more years of spawner abundance data are used to make the estimate, and the associated probability of extinction will decline.

The sampling error was determined in the same way that it is determined when estimating a mean from a number of samples. Estimating the geometric mean of CRR is equivalent to estimating the arithmetic mean of the natural logarithm of CRRs. For each possible value of the estimate reflected by that standard error, we calculated the probability of going extinct, then summed over all of the probabilities. We then included the effect of sampling error in this way, and the probability of extinction reflected in the delisting criteria was much greater than it would have been if the average CRR were known exactly (Appendix 3). From this analysis we determined that 9 samples of the CRR were needed to obtain an extinction probability less than 0.10. Because of the potential lag in reproduction of 4 years, this would require spawning abundance data from 13 years. While this result accounts for the error in estimating the future average CRR from spawner abundance, it is not usable because it assumes that spawner abundance will be known exactly. The error involved in estimating spawner abundance must also be accounted for.

Error in Estimating Spawner Abundance

The calculation of extinction rate assumes that spawning abundance is known exactly, which was very close to true from the mid 1960s until recently, but has not been true in the last several years, and will not be true in the future. When the gates of the RBDD were closed and migrating fish were force to use the fish ladder almost all were counted, yielding a precise estimate of number in the spawning run. Since 1986, the RBDD gates have been up during a substantial part of the run to improve adult fish passage conditions, and the precision of this estimate has declined substantially. Counting spawners over weeks 20 through 32 instead of the complete run (Figure IV-3) leads to a regression estimate (in logarithms) with a standard error of 0.831 which implies that the standard error limits will range from 43% to 230% of the estimate.

In the future, spawning escapement run-sizes will be estimated either by counts at RBDD or another method. To assess the effects of the error involved in this estimation, we first calculated the amount by which the standard deviation of the error in estimation of CRRs would be increased by not knowing the spawning abundances exactly. Next, we computed the way in which the probability of extinction changed as the standard deviation in that estimation error varied. As the standard deviation of the estimation error in estimating spawning abundance increased, the probability of a population that satisfied the extinction criteria going extinct increased. Thus, as the error in estimating spawning abundance increases, we need to choose a greater number of samples to guarantee that the probability of extinction in 50 years will be less than 0.10.

Currently the only available method for estimating spawning abundance is from counts during weeks 20 through 32, rather than for the entire duration of the run. Using this estimation method would require about 18 samples to reduce the extinction probability to less than 0.10. Since a sample of CRR can be obtained only after 4 years, this would imply 22 years of sampling. On the other hand, if a new method could be developed with a 25% error, we could achieve an extinction probability less than 0.1 with only 9 samples (13 years). A standard error of 25% was chosen because it is within the achievable range for methods of estimation of abundance in general.

On this basis, we specified that the geometric mean of cohort replacement rate must be 1.0 based on 9 samples (or 13 years), assuming that the estimation error is going to be roughly 25%. If that precision cannot be achieved, then the number of samples of CRR required to estimate the average CRR will have to increase by 1 sample for every 10% increase in relative error. This is the amount by which sample size appears to have to be increased to maintain an extinction probability of 0.10 (Appendix 3).

In addition, this analysis uses the variation in actual estimates of CRR for winter-run chinook to estimate the range of variation in the natural cohort replacement process, which is a key element in the simulation and viability analyses. The observed rates, particularly since 1986, also include measurement error. The viability analysis includes assumed, additional measurement errors in simulation runs. This has the potential to double count measurement error and overestimate the time required for delisting. However, in fact, the double counting has only a negligible effect on delisting time.

Genetic Considerations

In addition to determining the abundance levels needed to reduce the probability of extinction to safe levels, we also evaluated the impacts of various abundance levels on the genetic composition of the population. The genetically effective population size, N_e is used in the management of genetic resources of endangered species to convey information about expected rates of inbreeding and genetic drift, which can affect fitness and adaptive potential (Hedrick and Miller 1992). Several minimum effective population sizes (including males and females) have been proposed in the conservation genetics literature $N_e = 50$ as a lower limit to avoid inbreeding depression (Franklin 1980); $N_e = 500$ to avoid long-term loss of genetic variation (Franklin 1980, Lande and Barrowclough 1987); and $N_e = 5{,}000$ to maintain potentially adaptive variation for the long term (Lande 1995). In the absence of data on inbreeding depression and genetic drift, however, these limits provide only general guidelines for maintenance of genetic resources (see

Hedrick and Miller 1992).

With respect to the recovery of the Sacramento River winter-run chinook salmon, there are two genetic issues of concern: 1) the effects of past and present reductions in population size on population fitness and population growth rate, and 2) the genetic consequences of meeting the delisting criteria proposed in this plan. For the years 1991, 1992, and 1993, minimum estimates of the yearly overall effective size of the winter-run population are 21.9, 127.3, and 39.0, respectively, and the maximum estimates are 61.1, 401.0, and 108.6, respectively (Hedrick et al. 1995; see Chapter 5, Goal IV). Because this species is semelparous and the average age at spawning is three years, the total effective population size for these runs can be approximated by summing the yearly estimates, yielding 188.2 as the lower bound and 571.2 for the upper bound. Both estimates are above the limit of 50 proposed to avoid inbreeding depression, but only the upper bound exceed the limit of 500 suggested to avoid long-term loss of genetic variability. Both numbers are far below the limit of 5,000 suggested as necessary to retain potentially adaptive variation. Thus, it is possible that the winter-run chinook population has already been genetically impacted by reductions in abundance, such that during recovery, it may not be possible to achieve historical rates of population growth.

The recovery criteria developed from the demographic extinction model appear to provide a genetically effective population size that is large enough to retain sufficient genetic variation for maintaining present fitness and provide for future adaptability to changing environments. The genetic consequences of meeting the abundance criterion for delisting were evaluated assuming the lower (0.1) and upper (0.333) bounds on the ratio of effective spawning abundances to spawning abundance remain the same as the range of values for the few estimates made for wild fish, 0.1 and 0.333 (Hedrick et al. 1995). Using these values, upper and lower bounds on the effective number of spawners when the number of spawning females is 10,000, are 2,000 and 6,666, respectively. Summing over three years provides a rough approximation of N_e per mean generation, which yields a lower bound of 6,000 and an upper bound of 19,998. Again, both bounds exceeds the recommended level of 500 for long-term maintenance of genetic variation and the 5,000 proposed to retain adaptively relevant genetic variation for the long term.

Figure IV-1. Cohort replacement rates estimated from spawning abundance counts at Red Bluff Diversion Dam. (Note that they have rarely been greater than 1.0, the value required for a self-sustaining population in a constant environment.)

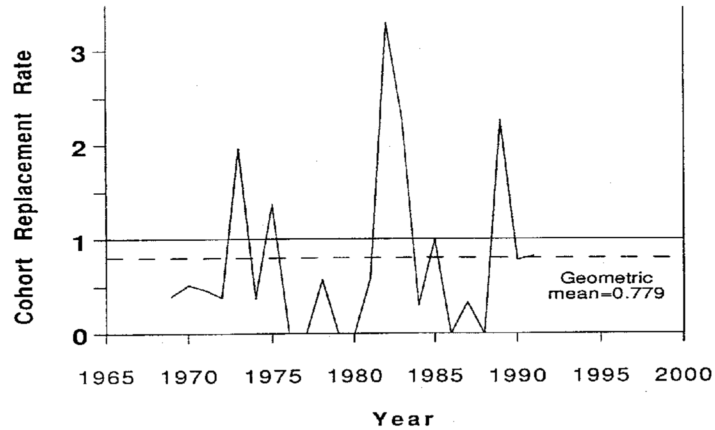


Figure IV-2. The fit of the model to Spawning abundance data, that determined the value of Cohort Replacement Rate (CRR) in Figure IV-3.

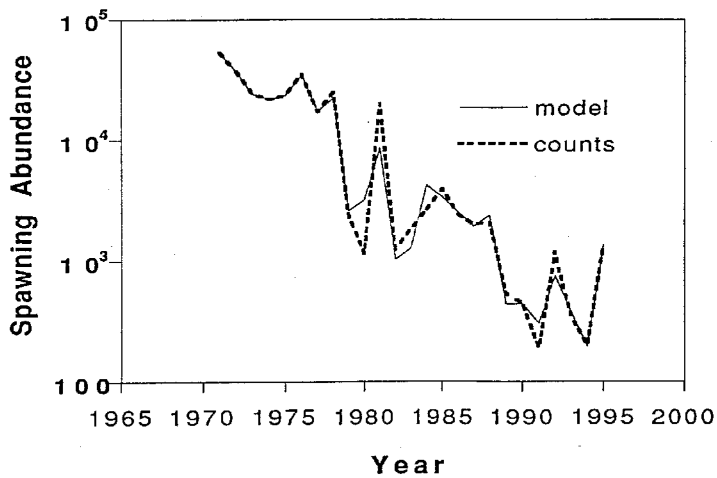
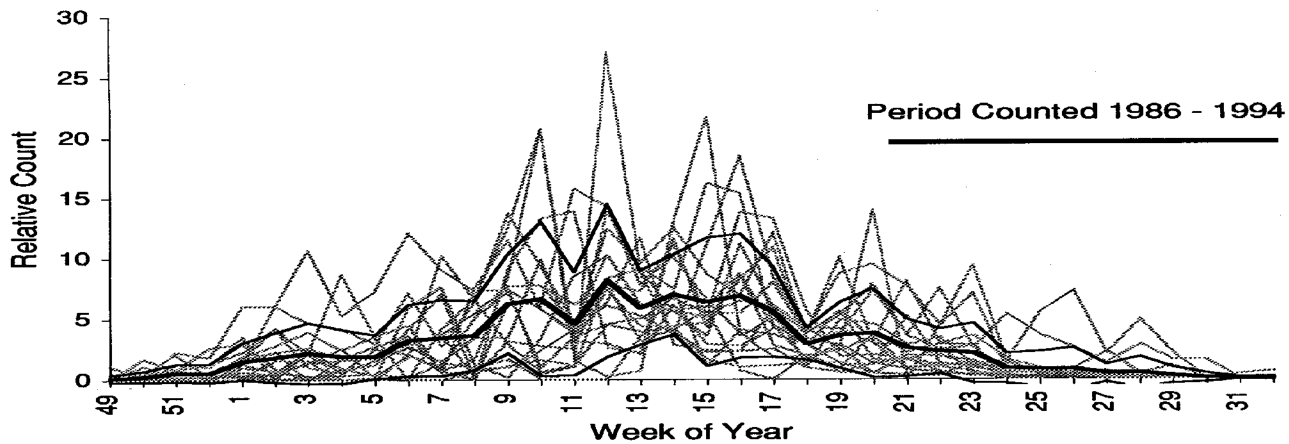


Figure IV-3. Spawning abundance counts at the Red Bluff Diversion Dam from 1967 to 1985. (The darkest line is the weekly mean, the lighter two lines are the weekly mean plus and minus one standard deviation, and the lightest lines are the actual data. Note that estimates from the 1986 to the present are based on a small fraction of the run.



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